Non-native Invasions Following Fire in Southwestern Colorado: Long-term Effectiveness of Mitigation Treatments and Future Predictions

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TABLE OF CONTENTS

EXECUTIVE SUMMARY

Six large, severe fires have burned tens of thousands of hectares in Mesa Verde National Park (MVNP) during the last 15 years, and extensive portions of those burned areas have been invaded by non-native plant species. Fire and weed invasion are primary management concerns not just in MNVP, but in similar piñon-juniper and mountain shrubland vegetation types throughout the Colorado Plateau. Aggressive non-native species may competitively displace native species, including endemic species and others of limited geographic distribution, and may alter fundamental ecosystem processes such as biogeochemical cycling, plant competitive interactions, and disturbance regimes. The potentially most threatening weed species in MNVP -- the ones of greatest management concern -- include muskthistle (*Carduus nutans*), Canada thistle (*Cirsium arvense*), and cheatgrass (*Bromus tectorum*). Many other non-native but apparently less aggressive species also are of secondary concern, e.g., prickly lettuce (*Lactuca serriola*), Russian-thistle (*Salsola iberica*), and alyssum (*Alyssum parviflorum*). In this study, we investigated the patterns and processes of weed invasion following the 1989, 1996, and 2000 fires in MNVP, to identify the kinds of plant communities that are most vulnerable to post-fire weed invasion, and to evaluate the effectiveness of a variety of weed mitigation methods including aerial seeding of native grasses, chemical eradication, and mechanical eradication. We summarize each of these two components of our study separately in the paragraphs below.

<u>Patterns & Mechanisms of Vulnerability to Weed Invasion</u>

To evaluate patterns and mechanisms of weed invasion, we conducted five related studies. First, we re-visited 121 sampling points within the 1989 Long Mesa burn where species composition had been inventoried in 1991, and recorded the non-native species still present at each point in 2002. Second, we re-visited 80 sampling points in the 1996 Chapin 5 burn that had been established in 1997, to record composition and density of non-native species still present in 2002 and 2003, and also established 25 control plots in unburned areas that were otherwise similar to the burned sites. Third, in 2002 we sampled species composition and density in 100 new plots located within the 2000 Bircher and Pony fires, using two different sampling methods: an inventory of all species within a circular 1000-m^2 plot (the "survey method"), and a multi-scale sampling of nested 1-m^2, 10-m^2, 100-m^2, and 1000-m^2

plots (the "Modified-Whitaker" method). Fourth, we collected soils from within the 2000 Bircher and Pony fire perimeters, and from unburned but otherwise similar areas nearby, and measured soil physical properties in 2002 and nutrient concentrations in 2002 and 2003. Finally, we collected soils from burned and unburned stands representing the major vegetation types in MNVP, and determined the composition of the soil seed-banks.

Broad-scale patterns of vulnerability to weed invasion: The relative vulnerability to weed invasion of each major type of habitat in MNVP can be predicted from basic vegetation and soils characteristics. We first describe the major patterns, and then provide the supporting data below. At the scale of the entire park, mature or old-growth piñon-juniper woodlands growing on Mikim loam and Arabrab-Longburn soil types are most vulnerable to post-fire weed invasion. This is partially because mature piñon-juniper woodlands tend to have a relatively sparse ground cover of native herb and shrub species capable of rapid post-fire sprouting, and partially because Mikim loam and Arabrab soils have as-yet-undefined soil characteristics that apparently are very favorable for weed growth. (These two soil types also support the greatest abundance of weeds on unburned sites) The second most vulnerable habitat is mature and old-growth piñon-juniper woodlands growing on other soil types. The sparse growth of native plants immediately after fire in piñon-juniper woodlands means that space and nutrients are readily available to support the establishment of non-native weed species (which generally require abundant light and nutrients). The soil seed bank under all kinds of piñon-juniper woodland tends to contain all of the major invasive species, even where adult weed plants are rare or absent. The presence of this weedy seed bank within undisturbed piñon-juniper woodlands, coupled with the potential for seed dispersal from nearby disturbed areas where adult plants are relatively abundant, and the open, nutrient-rich post-fire environment, means that nearly all piñon-juniper woodlands in the park are very susceptible to rapid and extensive weed invasion following fire (or other kinds of disturbance). Once established, the most aggressive of these non-native species may persist for at least 13 years (and probably much longer).

The least vulnerable type of vegetation at the scale of the entire park is mountain shrubland. This is because most of the dominant shrubs and herbs of the mountain shrub community are capable of rapid and prolific sprouting after fire – even where fire kills all above-ground vegetation and consumes most of the organic soil layer. The sprouting native species apparently take up available space and utilize available nutrients so quickly and efficiently that non-native species have limited opportunity to become established, even though weed seeds are present in the local soil seed-bank and also disperse from sources outside the burned area. Riparian vegetation, and other plant community types that contain a mix of sprouting and non-sprouting species, are intermediate in their vulnerability to non-native invasion. For all vegetation types, weeds are consistently more abundant in burned areas than in nearby unburned stands. Weeds also tend to be most abundant in years of normal or high precipitation, and are less abundant in extreme drought years like 2002.

The broad-scale patterns just described are supported by the data collected in this study. In 2002, 13 years after the 1989 Long Mesa fire, the average number of non-native species within each 314-m^2 plot was 2.2 for burned piñon-juniper woodland, but only 0.4 for burned mountain shrubland. In 2003, 8 years after the 1996 Chapin 5 fire, average density of invasive species (number per 310-m^2 plot) ranged from 1008 in piñon-juniper woodlands, to 24 in riparian areas, to only 9 in mountain shrublands. Within the perimeter of the Bircher and Pony fires of 2000, average density of invasive species (number per 310-m^2 plot) in 2003 (a year of somewhat below-average precipitation but *not* extreme drought) was 724 - 918 in burned areas but only 38 in unburned areas, while in the extreme drought year of 2002 the average density was 6 in burned areas and 0 in unburned areas. Also within the perimeter of the Bircher and Pony fires of 2000, average density of invasive species (number per 310-m^2 plot) in 2003 was 3650 and 1850 on Mikim loam and Arabrab-Longburn soil types, respectively, but was <200 on all other soil types. Even on unburned sites, the Mikim loam and Arabrab-Longburn soil types supported an average density (number per 310 m^2) of 330 and 20, respectively, while density in unburned sites was <5 for all other soil types. Burned and unburned areas differed substantially in several important soil properties, all of which support the idea that nutrient conditions are generally more favorable for plant growth (especially for weeds) in burned than in unburned areas. Compared with nearby unburned areas, soils from burned areas within the 2000 Bircher and Pony fire perimeters exhibited higher available phosphorus and conductivity (a general measure of the concentration of dissolved ions) in both 2002 and 2003, and higher pH, available nitrogen and soil moisture in 2002 only.

Fine-scale patterns of vulnerability to weed invasion: The average values reported above for broad-scale patterns of weed density were accompanied by very high standard errors, indicating that a great deal of additional variability exists at fine scales within each major vegetation type and between burned and unburned sites. At the scale of a 1000-m^2 plot, a greater number (richness) of native plant species was strongly correlated with a greater number of non-native species, indicating that habitats that support high native biodiversity are at the greatest risk of weed invasion. This pattern was essentially the same with both sampling methods: r=0.65 with the survey method; r=0.54 with the "Modified-Whitaker" Method. However, the "Modified-Whitaker"

Method provided an additional important piece of information: the correlation between native and non-native diversity is weaker when plot size is reduced to 100 m^2 (r=0.32), and disappears altogether at plot sizes smaller than 100 m^2. Some other factor(s), not measured in this study, apparently control(s) weed invasion at the micro-scale (<100 m^2).

Weed density in 2002 also was correlated with local soil properties at the scale of a 1000- m^2 plot, both within the perimeter of the 2000 Bircher and Pony fires and in nearby unburned areas In burned areas, weed density (number per 310-m^2 plot) was greater in places having greater soil nitrate and a higher percentage of silt than in places where nitrate and % silt were lower. In unburned areas, weed density increased with increased soil nitrogen, phosphorus, and conductivity. These findings further support the idea that weeds thrive in the high soil nutrient conditions that typically characterize post-fire environments and that also tend to support higher native plant diversity in either burned or unburned areas.

Effectiveness of Weed Mitigation Measures

Following the 1996 Chapin 5 fire, a variety of weed mitigation techniques were applied to sites considered to be at the greatest risk of weed invasion. These methods included aerial seeding of native grass species in the fall of 1996, and chemical and mechanical eradication of localized thistle populations in 1997, 1998, and 1999. Eighty sampling points were established in 1997 to monitor treatment effectiveness, and we re-visited these sites in 2002 and 2003 to document changes in native and non-native plant composition and density.

In 2003, average weed density was much greater in burned areas (that received no treatment) than in nearby unburned areas. However, muskthistle density was lower by two-thirds, and native grass density was higher by 100-fold, in the areas that received aerial seeding treatments than in areas that received no treatment. These results indicate that aerial seeding of native grasses *did* reduce the density of post-fire weeds and *did* increase the native component of the post-fire plant community. However, aerial seeding *did not* completely prevent weed invasion; seven years after the fire, the non-native species were still present throughout the area that burned in 1996, although their densities were significantly lower where the aerial seeding treatment had been applied.

Chemical treatment of Canada thistle (spraying with the herbicide Curtail) in 1997 (one year post-fire) and 1998 (two years post-fire) resulted in almost complete mortality, such that Canada thistle was nearly absent from treated areas in 2003.

Mechanical treatment of muskthistle (removing seed-heads and digging out plants with a shovel) was effective only in areas where native grasses were present; muskthistle was absent from 10 such sites in 2003. However, in other sites where native grasses were not present, muskthistle was still in evidence in 2003, despite mechanical treatments in 1997. Both Canada thistle and muskthistle have persisted in burned areas that were not treated by any means. However, two of the less aggressive weed species, Russian-thistle and prickly lettuce, both of which were abundant in the Chapin 5 burn in 1997 and 1998 (one and two years post-fire), had almost disappeared by 2003 from treated and untreated areas alike.

Overall, our analysis of the various weed mitigation measures that were applied after the 1996 Chapin 5 fire, suggests four general conclusions that may be applicable to future weed mitigation efforts. First, aerial seeding of native grasses is very effective at reducing weed densities and rapidly increasing the native grass component of the post-fire plant community, but seeding does not prevent weeds from becoming established in burned areas. Continued monitoring of our permanently marked sample points will be needed to determine whether or when the weeds eventually disappear from seeded and unseeded areas. Second, chemical treatment is very effective at eradicating Canada thistle; however, because it is labor-intensive, it probably is feasible only in localized areas. Third, mechanical removal is effective at eradication of muskthistle, but only where native grasses are present. Like chemical treatment, mechanical eradication is labor-intensive and probably is not feasible over large areas. Finally, some of the less aggressive weed species may largely disappear within less than a decade even without treatment.

Cheatgrass – a Significant and Increasing Threat to Ecological Integrity in MNVP

Our weed studies prior to 2003 focused on the most abundant and aggressive non-native species that we found in burned areas – notably muskthistle and Canada thistle. Prior to 2003, cheatgrass was not a major component of post-fire weed communities, nor was it common anywhere on the mesa tops in MNVP. Indeed, it was thought that the relatively high elevation, high summer precipitation, and cool temperatures that characterize MNVP, would prevent cheatgrass from becoming important within the park -- even though cheatgrass has spread prolifically in many surrounding areas at lower elevations where the climate is warmer and drier. However, in 2003 dense stands of cheatgrass appeared in many places throughout MNVP where it had not been present (or at least not conspicuous) previously. New cheatgrass locations included areas at surprisingly high elevations, e.g., at ca. 2480 m elevation along the Moccasin Mesa road, where the Bircher fire had burned in 2000. Equally alarming was our finding of no significant difference in cheatgrass density in 2003 between areas on Park Mesa that had been aerially seeded after the 1996 Chapin 5 fire and nearby areas that had received no treatment.

We regard this rapid proliferation and spread of cheatgrass in MNVP as a very serious threat to the long-term ecological integrity of park ecosystems. Cheatgrass invasion elsewhere in the West has led to profound changes in native plant species diversity, community structure, and fire regime. A winter annual, cheatgrass grows rapidly during late winter and early spring, potentially consuming much of the soil moisture needed by later-growing native plants. When dense stands of cheatgrass die and cure in early summer, they provide a continuous bed of highly flammable fuel which can readily carry a relatively low-intensity but fast-moving fire. If cheatgrass continues to spread into recently burned areas in MNVP, it may cause a switch from the previous fire regime of infrequent fires that occurred only during extremely dry periods, to a new fire regime of frequent fires. Because the native flora is adapted to the historical fire regime, a change of this kind could produce rapid and irreversible degradation of MNVP's native vegetation. Given the potential magnitude of this threat, we make two recommendations: first, activities that promote the spread of cheatgrass, e.g., any disturbance of the soil or native vegetation, should be discouraged; and second, cheatgrass distribution and abundance in the park should be monitored and evaluated periodically.

INTRODUCTION

Large wildfires have impacted vegetation dynamics in the uplands of southwestern Colorado throughout history. We have detected fire intervals of about a century in the mountain shrublands (petran chaparral) and four or more centuries in the mature piñon-juniper woodlands (Floyd et al 2000, 2004, Romme et al 2003). Prehistoric Ancestral Puebloans of the Mesa Verde region very likely used fires to clear woodlands and maintain agricultural fields (Wycoff 1977) and historic accounts describe intentional fires set by Ute Indians (M. Colyer oral histories). It is unlikely however that human uses of fire were extensive or common enough to significantly alter fire dynamics (Allen 2003). Despite frequent lightning events in the summer monsoons, fires in these upland areas were characterized by numerous small, single-tree fires and much longer intervals between large fires relative to other southwestern uplands (Allen 2003). The long intervals allowed for development of extensive old-growth piñon-juniper woodlands (Romme et al. 2003); coupled with exclusion of cattle grazing since about 1930, these woodlands now support a rich understory flora (Floyd 2003).

However, the frequency of large-scale fires has increased in the last fifteen years, in part due to regional drought conditions preceded by several wet decades (Floyd et al. 2004). Eight large wildfires have burned during the last 100 years on Mesa Verde, a prominent cuesta that rises above the lowlands near the Four Corners junction of Colorado, New Mexico, Arizona, and Utah (Figure 1). Six of those fires have occurred in just the last 15 years (in 1989, 1996, 2000-Bircher Fire, 2000-Pony fire, 2001, and 2003). The current landscape differs also from the past in trends of floristic responses to the fires. Native plant species in Mesa Verde are now accompanied by a suite of recent arrivals, including *Carduus nutans*, *Cirsium arvense*, *Lactuca serriola*, *Bromus tectorum*, *Salsola kali*, *Lappula redowskii*, *Ranunculus testiculatus*, *Alyssum parviflorum*, and *Lepidium latifolia*. As part of our evaluation of post-fire landscapes, we focused on regeneration and spread of native and non-native plant species after recent fires in Mesa Verde National Park (MVNP) and surrounding Ute Mountain Ute (UMU) Tribal lands.

A contentious debate surrounds impacts of biodiversity on ecosystem functioning and stability, especially with regard to the system's ability to resist invasion by exotics (Dukes 2000, Higgins et al., 1999, Hughes and Petchey 2001, Grime 1998, Levine 2000, McCaan 2002). Community stability, the life history of natives and invasive species, fertility, and disturbance patterns are among those attributes whose variation influences invasibility (Prieur-Richard and Lavorel 2000). Recent studies contrast the role of species richness in either protecting communities against invasion, supporting the "diversity-resistance hypothesis" originally proposed by Elton (Elton 1958, Kennedy et al 2002, McCann, Kevin Shear. 2000) or, conversely, other studies document a pattern of increasing invasibility with increasing species richness (Levine 2000, Stohlgren et al 2001, Higgins et al 1998). These seemingly opposite trends may reflect the scale of the study (Hughes and. Petchey 2001, Kennedy et al 2002). Neighborhood analyses support a positive relationship between species diversity and resistance (Kennedy et al 2002), or functional diversity and resistance (Dukes 2001), while regional or landscape scales often reflect a negative relationship between species diversity and resistance. In the latter, more diverse communities are targets of increasing levels of invasion (Higgins et al 1999, Levine 2000,Stohlgren et al 1999, Stolgren et al 2001). For example, in a series of multiscale (up to 1000 m^2) or large (672 m^2) plots in a variety of habitats across eight states, greater native diversity was accompanied by greater non-native diversity (Stohlgren et al 2003). Regardless of the relationship between biodiversity of natives and invasives, it is clear that species richness alone cannot account for inter-community differences in invasibility (Prieur-Richard and Lavorel 2000). In fact, species richness has become discounted as the key parameter that dictates invasibility, while diversity and characteristics of functional groups or trophic levels are often deemed a more important deterrent of invasives (Dukes 2001, Lehman and Tilman 2001, Tilman et al 1997). Few studies illuminate mechanisms that allow particular species to invade particular communities. It is important that confounding variables impacting natives and invasive species, including propagule dispersal (Levine 2000), herbivory (Prieur-Richard et al 2002), spatial patterns in the invaded landscape, and fertility (With 2001) be considered.

The role of disturbances, including fire, is of unequivocal importance in this debate (Prieur-Richard and Lavorel 2000) yet few studies address the relative importance of natives and invasives after wildfires. Here, we consider both the possible role of disturbances created by large wildfires in invasibility in a semi-arid woodland ecosystem in southwestern Colorado and the impact of the invasions on maintaining biodiversity in early post-fire succession.

In order to gather data on potential invasive species and on characteristics of habitats that are particularly invasible following fires, we documented the diversity and density of species returning to different plant communities within the burned landscapes in Mesa Verde. Two years after the year 2000 fires, we used two methods, comparing their efficacy in addressing the questions listed below. We also investigated the contributions of the soil environment (seed bank storage, texture, and chemistry) in determining patterns of non-native invasion. Finally, we evaluated post-fire mitigation activities.

We addressed the following questions:

- What is the association between native and non-native plant diversity in burned and unburned landscapes in Mesa Verde?
- Do characteristics of functional groups in the pre-fire community influence invasion pattern?
- Does the scale of sampling influence our ability to detect associations between native and non-native species?
- What is the contribution of the seed bank to early successional communities and invasive patterns after fire?
- What is the contribution of microhabitat differences in soil type, texture, and chemical properties to invasibility?
- How can we best model the invasibility of communities in Mesa Verde after fire for future predictions?

- How effective are aerial seeding of native grasses, mechanical, and chemical treatments in preventing non-native invasions seven years after treatments in piñon-juniper woodlands?

Study Area

Mesa Verde is located in the southwestern corner of Colorado, USA (Figure 1). Mesa Verde National Park (MVNP) occupies the middle of a prominent cuesta that slopes from 2060 m in the south to 2485 m in the north. The south and eastern edges of the cuesta are bordered by the Mancos River, while steep escarpments characterize the north. The cuesta is composed of Cretaceous sandstones and shales.

Pinus edulis (piñon pine) and *Juniperus osteosperma* (Utah juniper) dominate the lower elevations of the mesa tops and canyon slopes. *Purshia tridentata* (bitterbrush) is often the dominant understory shrub, other species may include *Quercus gambelii* (Gambel oak), *Amelanchier utahensis* (Utah serviceberry), *Artemisia tridentata* (big sagebrush), *Peraphyllum ramossisium* (squaw apple), and *Fendlera rupicola* (fendler bush). Common forbs include *Penstemon linearoides, Pedicularis centranthera, Cryptantha bakerii, Polyonum sawatchensis, Lupinus ammophila, Astragalus scopulorum, Calochortus nuttallii, Commandra umbellata, Cymopterus bulbosus, C. purpurescens*, and *Yucca baccata.* The dominant understory grass is *Poa fendleriana* (mutton grass). (For a complete description of flora typical of other piñon-juniper types, see Chapter 3, Floyd, 2003).

Piñon-juniper woodlands in Mesa Verde tend to be old and structurally diverse. As the woodland matures, trees that succumb to insect and fungal pathogens and natural senescence create canopy openings. Small, often single-tree fires are an annual occurrence, leaving openings in woodland canopy. Black stain root disease and engraver beetles kill small clusters of trees, creating openings in the canopy. The resultant heterogenous canopy structure can support a high diversity of herbaceous species in years of normal precipitation; in 400 sampling areas, the number of understory forbs and shrubs was the same in open and dense canopies (averaging 12 species per 100 m2), with a slightly greater number (16 species per 100 m2) in the intermediate canopies (Floyd-Hanna et al 1993, Floyd et al 2003).

It is important to consider disturbances that were evident before fires because invasive plant species were well established in these sites which then serve as foci for invasions. Construction of roadways brought muskthistle, perrperweed, and other invasives into the Park. A sewer line was cut through Chapin Mesa in the 1990's, disturbing old growth woodlands. The Headquarters area, including a CCC camp, a million gallon water tank, and several sewage facilities harbored invasive species. Much of the Bircher fire in 2000 reburned the 1959 Morefield fire that had been seeded with *Bromus inermis* (smooth brome), an invasive grass, and that species was well established throughout Morefield and Prater Canyons. The Bircher fire also reburned a portion of the 1972 Mocassin Mesa fire in which *Agropyron intermedium* (intermediate wheatgrass) and muskthistle had become established. Both of these invasive species were therefore immediately available to invade areas burned in the Bircher fire in 2000. Also, both smooth brome and intermediate wheat grasses are rhizomatous and likely excluded other species, including other invasives.

Six large wildfires have occurred on Mesa Verde in the past fifteen years. In this study, we focused only on the 1989 Long Mesa fire, the 1996 Chapin 5 Fire, and the 2000 Bircher and Pony Fires. The Long Mesa Fire began with a lightning strike in a dense, old-growth piñon-juniper woodland and burned 1800 ha of mesa-top, deep canyons and steep north-facing habitats with piñon-juniper, mountain shrublands, and Douglas-fir vegetation. No mitigation treatments were applied to this fire. The Chapin 5 fire in August 1996 burned 1972 ha impacting seven vegetation types. Under Burned Area Emergency Rehabilitation (BAER) funding, mitigation treatments—aerial seeding, herbicide, mechanical, and biological-- were applied. The long-term effectiveness of the first three treatments after the Chapin 5 fire are documented in Part II of this report (biological controls were applied by Dr. Deb Kendall, and are reported elsewhere). The Bircher fire burned 10,592 ha in late July 2000, including piñon-juniper woodlands of varying stages of succession, mountain shrublands and open wet meadows. A few weeks later, the Pony Fire burned 2,272 ha., consuming large tracks of old-growth piñon-juniper woodlands. In Part II of this report we focus on post-fire recovery and short-term effectiveness of mitigation following the Bircher and Pony Fires.

Part I. Characteristics of invasible habitats after fire

Methods

Long Mesa 1989 Fire:

Two sampling grids (originally established in 1991 to include the range of pre-fire vegetation on Long Mesa and within Long Canyon (Floyd-Hanna and Romme, 1991) were revisited in 2002 to evaluate the persistence of non-native invasive plant (Figure 2). The "Southern Grid" consists of a square kilometer, sampled systematically every 100 meters, for a total of 121 sampling points. This grid represented the heterogeneity of vegetation within the piñon-juniper/shrubland mosaic that characterizes low to mid elevations in MVNP. The "Northern Grid" was placed within the essentially treeless, mountain shrub community which extends over much of the northern and slightly higher sector of MVNP. As the vegetation was relatively homogenous, the size was reduced to 1 km by 0.6 km and the number of sampling points to 66. All points were severely burned. "Control points" consisted of vegetation missed by the 1989 fire. During the present study we returned to a subset of these points and recorded presence or absence of each invasive species within a 10 m radius of the point.

Chapin 5 1996 Fire:

The Chapin 5 Fire was separated into 19 "habitats" by stratifying vegetation communities, substrates, and slope categories in IDRISI GIS. Random sample locations were created within each habitat. A total of 80 burned points were sampled in 1997, 1998, and 1999 with Burned Area Emergency Rehabilitation (BAER) program funding. During the current study we returned to these burned points and also established 25 "control" points in nearby unburned habitats on Chapin Mesa. We visited all 105 points in 2002 and 2003 to evaluate long-term trends in native and invasive plant distribution (Part I) and to assess the success of mitigation treatments (see Part II below). At each point the density of each non-native plant species that was singly rooted was counted in each of three 310 m^2 plots. The density of rhizomatous species (such as cheatgrass) were estimated.

In 1997, our research group created a simple spatial model for possible non-native susceptibility in Mesa Verde by overlaying GIS layers with pre-fire vegetation, distance to known non-native sources, and prominent wind vectors. This simple risk model was extremely helpful in the BAER assessments, because it allowed us to predict general areas of "high-risk," "moderate-risk" and "low-risk" to non-native invasions would occur. One of the goals of the current project is to refine this preliminary model, including additional characteristics of residual vegetation (perennial sprouters and germinants from the seed bank) and soil fertility.

Bircher and Pony Fires:

One hundred points were randomly identified using a "sample" procedure in the GIS software IDRISI; a control point of similar slope, aspect, pre-fire vegetation and geologic substrate was paired with each randomly generated burned point (Figure 2). Data were collected during May-August 2002. Native and invasive species diversity was assessed by two methods at the same locations. At each point, the "survey" method was used to assess biodiversity; we thoroughly surveyed a 1000 m^2 plot (18 meter radius from the point) and recorded each native and non-native plant species present. The second sampling method was the "“Modified-Whitaker” " system of embedded small and large plots (Stohlgren et al 1997). This method has been advocated in landscape-scale investigations of plant invasions (Stohlgren et al, 1999). Fifty-four points were selected from among the 100 points. At these points, Modified-Whitaker multi-scale 1, 10, 100 and 1000 m^2 plots were established to determine species richness. We recorded the number of native and non-native species in each plot as well as the cover of each species in the ten 1 m^2 plots (Stohlgren 1997). Thus, 100 points were surveyed by the "survey" method and 54 points were surveyed by both methods in 2002. We returned to each point in 2003 and recorded the density of each singly-rooted non-native species in three 314 m^2 plots. Densities of rhizomatous species were estimated.

The densities of native and non-native species collected in either the "survey" or the "Modified-Whitaker" procedures were subjected to correlation analyses (Sokal and Rolf 1977) using SPSS software. Pearson correlations were run on the entire "Modified-Whitaker" data set, then repeated for each plot size. Burned and unburned sample points were analyzed separately using both the "survey" and the "Modified-Whitaker" data sets.

Soil chemical analyses were conducted on soils collected in 2002 and 2003, whereas soil physical properties were measured only on soils collected in 2002. In order to refine the model of invasibility, we sampled soils among paired severely burned and unburned samples in the Bircher and Pony Fires of 2000. We tested:

- pH (in DI H2O)
- Soil organic matter (acid treated LOI)
- Available phosphorus (resin)
- Conductivity
- Texture: sand, silt, clay (hydrometer)
- Moisture (gravimetric)
- Available nitrate and ammonium (2M KCl)

- Net N mineralization (lab incubation)

Soil nitrogen, i.e. nitrate and exchangeable ammonium, was measured by preparing soil extracts with 2M KCl. The extracts were analyzed with an autoanalyzer. Organic matter was estimated by pre-treating samples with concentrated HCl to eliminate carbonate, i.e. inorganic carbon. This was followed by ignition at 550°C and loss of organic material was calculated. Soil pH was measured electrometrically following the protocol outlined by Robertson et al. (1999). Phosphorus was measured by laboratory resin extraction and the Murphy and Riley Procedure (Lajtha et al. 1999). Soluble salts were determined by electrical conductivity analysis following the protocol of Janzen (1993). Texture, i.e. soil particle size distribution, was measured by hydrometer (Sheldrick and Wang 1993). Water content was measured gravimetrically.

At the same sites that chemical analyses were run, samples of the top 10-15 cm of soils were collected for seed bank analysis. Samples were "floated" to remove organic matter, and sieved with multiple pore sizes (Meredith Matthews, personal communication). Seeds were collected from each fraction and the species was identified using reference seed collections at San Juan College, Farmington, New Mexico, and the Anasazi Heritage Center, Cortez, Colorado.

RESULTS

Testing of Sampling Procedures on Bircher and Pony Fires

The average number of species is indistinguishable using the "Survey" or ""Modified-Whitaker" " method in 2002 (Table 1). The time invested in each ""Modified-Whitaker" " plot is far greater-- it took about 3 hours per point to sample the three sizes of embedded plots (ten 1 m^2, 2 two 10 m^2, one 100 m^2, and one 1000 m^2). The circular plot surveyed in the "Survey" method was generally finished in less than an hour.

Table 1. The number of plant species recorded in plots of varying size sampled by "Survey" and ""Modified-Whitaker" " methods at the same points. Data are shown as mean, standard error. n = number of plots. Survey plots were 18 m radii, and only the 1000 m^2 Modified-Whitaker plots were used in this analysis.

Sampling Type	Number of Species	n
"Survey"	22, 1	54
Modified-Whitaker	23, 1	54

What is the association between native and non-native plant diversity? Using either sampling method, strong positive correlation coefficients suggest that habitats supporting increasingly greater numbers of native species also maintain an increasingly diverse non-native flora (Table 2).

Table 2. Results of correlation analyses of the number of native and number of non-native species in sample plots throughout Mesa Verde National Park. The significance of the correlation coefficients was tested with T-tests.

Sampling Type	Points sampled	Correlation Coefficient	Probability
Survey method	100	r = .648	p<.01
Modified-Whitaker	54	r = .543	p<.01

Does plot size influence our ability to detect associations between native and non-native species? The "“Modified-Whitaker” " sampling design allows us to compare the scale of sampling, from 1 m^2, 10 m^2, 100 m^2 and 1000 m^2. Using plot sizes below 100 m^2, we did not detect a significant association, either positive or negative, among the density of native and non-native species. Only when plot size is increased to 100 m^2 and larger is a positive, significant, correlation detected (Table 3).

Table 3. Comparison of correlation analyses of the number of native and the number of non-native species among the plot sizes used in "Modified-Whitaker" sampling, Mesa Verde National Park. Correlation analyses test for an association of the number of native and non-native species.

Plot size	Correlation Coefficient	p	n
1	r=-.042	p>.05	619
10	r=-.026	p>.05	123
100	r=.324	p=.01	60
1000	r=.534	p<.01	54

Is the pattern of association between native and non-native species similar in recently burned (early successional) and old, well-established (late successional) habitats? Burned and unburned plots were analyzed separately in this analysis. In both early successional post-fire habitats and unburned habitats, areas that support a high diversity of native species also support a high diversity of non-natives (Table 4).

Table 4. Correlation analyses on sample plots that burned in the 2000 Bircher and Pony fires. Unburned, paired plots had not burned for several centuries. Only the 1000 m2 "Modified-Whitaker" plots were used.

Sample method	Recently burned	n	Unburned	n
"Survey"	r = .67 p<.01	53	r = .63 p<.01	47
"Modified-Whitaker"	r = .42 p=.05	29	r = .74 p<.01	25

<u>Trends in Non-native Invasions Following the Year 2000 Fires</u>

In 2002, during extreme drought conditions, few non-natives germinated. While we were able to detect small significant differences in the density of invasive species in burned and unburned paired samples (Table 5), correlations using these data are weak because the magnitude of invasions is considerably lower than post-fire invasions under "normal" precipitation patterns. Precipitation levels documenting severe drought during the study are shown in Figure 3. In 2003, precipitation was below normal but greater than 2002, allowing us to compare the three treatments: unburned, burned (severely burned from BAER classification) and seeded with native seed mixes and burned but untreated (see Part II). While there was an overall significant difference among the three treatments, no difference between seeded and untreated burned treatments was detected in 2003 (Table 6). This contrasts a consistent trend documented after the Chapin 5 Fire in which the density of invasive non-natives (primarily muskthistle in Mesa Verde) was significantly lower in seeded than unseeded samples (Part II). We attribute the lack of significance to the regional drought that reduced germination of both seeded grasses and invasive non-native species over all treatments within the piñon-juniper habitat type.

The second year of the current study, 2003, brought an unexpected event. *Bromus tectorum* (cheatgrass) expanded rapidly from small foci within the park and the region as a whole. We had not considered cheatgrass in 2002 as it was not present in the study plots. However, the fall precipitation in 2002 allowed it to take a tremendous hold in both burned and non-burned landscapes. Our subsequent analyses have separated cheatgrass from the analyses both to foster year to year comparison within our study and to acknowledge that the post-fire response by this annual grass is fundamentally different from the biennial and perennial non-natives that we have been following.

Table 5. The density of invasives (number per 314 m^2 plot) the Bircher and Pony sampling points in 2002 during extreme drought conditions. Values given are mean, standard error.

Treatment	Non-native Density	Sample size
Unburned	0, 0	68
Burned (combined)	5.8, 1.8 *	192

*T= 3.8, p<.05

Table 6. The density of invasives (number per 314 m^2 plot) the Bircher and Pony Fires' sampling points in 2003 after less than normal precipitation conditions. Values given are mean, standard error.

Treatment	Non-native Density	Sample size
Unburned	37.5, 22.9	40
Burned, seeded	918.2, 307	35
Burned, unseeded	724.3, 361	40

F=3.0, p=.05, significant difference in non-native density across three treatments

Role of Sprouting Species in Invasions

We observed after the 1989 and 1996 fires that when mature and old-growth woodlands burn at Mesa Verde, they are more susceptible to non-native plant invasions than other vegetation communities (BAER Final Reports, 1998, 2002). Three types of piñon-juniper woodlands can be considered a community type, one that lacks perennial sprouters. Mountain shrubland vegetation types comprise a community type that has one or more prolific sprouting shrub species. Open meadows and wetlands have many sprouting grasses or sedges and, therefore, they are grouped together. Here we consider the relative importance of residual sprouters in invasiveness by comparing the density of invasives across communities with and without prolific sprouting shrubs in Table 7. Seven years after the Chapin 5 Fire, there were significantly fewer muskthistles and other invasive forb species in habitats characterized by sprouting perennial shrubs or graminoids; piñon-juniper community types were the most invasible (Table 7). All habitats except riparian were in "severely burned" areas (BAER Plan Chapin 5 Fire, 1996).

Table 7. The density of invasive species in 310 m^2 plots in various burned habitats demonstrate that presence of sprouting species have a significant impact on invasibility. Data were collected in 2003, eight years after the Chapin 5 fire. Data are mean, standard error.

Community types	Prevalence of sprouting shrubs	Non-native Density	Sample Size
Piñon-juniper	Non- sprouters	1008.2, 296.7	67
Mountain shrubland	Prolific sprouters	8.7, 3.8	44
Riparian	Sprouting grasses	24.0, 24.0	3

$F=3.9$, $p=.02$ significant difference in non-native density across functional groups

Long-term Trends Following Fires: Long Mesa After 13 Years

To determine further the role of sprouting residual vegetation in maintaining native communities after disturbances, we returned in 2002 to the 1989 Long Mesa Fire and re-sampled a subset of our original sample points. Samples were divided among community types: piñon-juniper types lacking perennial sprouters and mountain shrublands characterized by prolific sprouters. Because severe drought conditions reduced growth throughout the area, we simply recorded the presence and absence of native and non-native species rather than cover or density. Thirteen years after the fire, while the number of native species was not significantly different in these two groups (average of 15 in piñon-juniper and 16 in mountain shrublands), there was a significant difference in the diversity of invasive species (Table 8). Even after 13 years, there is significantly greater diversity in the invasive flora in the sites lacking sprouting competitive species; it is likely that measures of density (during normal precipitation conditions) would corroborate this trend. These data support the importance of functional groups in defining invasibility of burned habitats.

Table 8. The number of native and invasive species recorded in sample plots in community types (piñon-juniper lacking sprouting species) and mountain shrubland (prolific post-fire sprouting) in the Long Mesa Fire 13 years after fire. Data are mean, standard error.

Type	Including sprouters	Lacking sprouters	Statistical significance (p)
Native	14.6, 3.6	16.4, 4.3	n.s.
Invasive	0.4, 0.6	2.2, 1.2	$T = 5.3$, $p<0.01$
Sample size	29	45	

Soil Chemistry and Invasibility

As suggested by the large standard errors in Tables 6 and 7, some sites within the "high-risk" piñon-juniper burn areas are targets while others remain free of non-natives. Inputs following the Bircher and Pony Fires of 2000 significantly increased soil pH, salinity (conductivity), and the availability of nitrogen and phosphorus. As a whole these findings reflect that combustion of plant materials results in rapid mineralization and release of base-forming cations and other essential plant nutrients. The drought conditions that persisted at Mesa Verde between the fire season of 2000 and when the soils were sampled in 2002 prevented leaching of fire-released salts and nutrients. The winter of 2002-2003 was considerably wetter, which likely explains why there were fewer significant differences between severely burned and unburned soil characteristics measured during the following spring (Table 9).

Table 9. Soil properties that were significantly different in burned and unburned sites two and three years following the Mesa Verde Bircher and Pony Fires. Values in parentheses are standard errors.

Property	unburned	burned	p
Two years after fire			
Available nitrogen $NO_3^- + NH_4^+$ mg gdw^{-1} soil	27 (8)	61 (5)	.001
Available phosphorus µg gdw^{-1} soil	32 (7)	50 (5)	.03
pH	6	6.4	.003
Conductivity dS m^{-1}	0.19 (.03)	0.36 (.02)	<.001
% soil water	3.1 (0.3)	2.1 (.2)	.003
Three years after fire			
Available phosphorus µg gdw^{-1} soil	28 (4)	45 (5)	.02
Conductivity dS m^{-1}	0.11 (.01)	0.18 (.01)	<.001

While we measured numerous soil attributes that were altered by fire, non-native densities within the actual burned areas only correlated weakly with variation in soil nitrate (Table 10). Non-native densities in burned areas were also found to correlate with percent silt, which did not vary as a function of burning. Interestingly, in unburned sites, although non-native densities were considerably lower than burned sites (Table 6), their distribution did correlate strongly with soil available phosphorus and weakly with net N mineralization and salinity. Findings from this initial analysis of the effects of fire on soil characteristics and, in turn non-native densities, suggest that the availability of nitrogen and phosphorus, as well as soil textural class, should be considered as important indicators of site invasibility.

Table 10. Correlations significant at the probability < .10 level in unburned and post-fire soil properties with non-native densities. Sample size=57, data shown are for 2002. Significance tests were student t-tests, probability values are shown.

Property	Pearsons Correlation Coefficient	Statististical Significance (p)
Unburned		
Net N mineralization	r = .43	p = .01
Available phosphorus µg gdw^{-1} soil	r = .56	p = .01
Conductivity	r = -.40	p = .01
Post-fire		
Available nitrate	r = .26	p = .06
Percent silt	r = .29	p = .08

Extrapolation to Mapping Units- NRCS

There was a significant difference in the density of non-natives (excluding cheatgrass) across seven soil types defined by the USDA-NRCS soil maps in the Bircher and Pony fires ($F= 3.9$, $p = .004$). These data (Figure 4) suggest that within the piñon-juniper habitat, Mikim loam and Arabrab-Longburn soils are most invasible after the soils are burned. There is not apparently simply due to percent silt or available nitrate (Table 10); these values were not significantly greater than less susceptible soils in the area. However, burned Mikim soils were significantly higher in phosphorus than other soils sampled (available phosphorus in Mikim samples averaged 75 µg gdw^{-1} soil whereas the overall mean available phosphorus was 49 µg gdw^{-1} soil); this trend does not apply to Arabrab-Longburn soils which were low in available phosphorus (average 27 µg gdw^{-1} soil). While no significant difference in non-natives was detected across unburned soil types ($F = 1.5$, $p >.05$) Figure 4 suggests that the Arabrab-Longburn soils likely support non-natives even when unburned.

Seed Bank

Piñon-juniper soil samples contained a greater diversity of seeds (33 taxonomic groupings, including families Poaceae, Malvaceae and Compositae) than did mountain shrubland soil samples (8 taxonomic groupings) and Douglas-fir samples (3 taxomonic groupings). The most common native species after fires, *Chenopodium alba* and *Polygonum sawatchensis,* were found in the unburned samples, but at much higher density in the burned samples. This is consistent with early emergence of these species into the post-fire communities.

Invasive species *Carduus nutans, Lactuca serriola, Taraxicum officiale,* and *Tragopogon pratens* seeds were detected in both unburned and burned samples. Their presence in unburned soils suggests that these seeds might germinate rapidly in the post-fire soils and therefore establish quickly. The density of muskthistle seeds was over ten times greater in the burned/unseeded plots than in the unburned plots in 2002, two years after the fire. Our data do not support conclusive trends that would define the relative contribution of dispersal or germination from seed banks to post-fire invasive populations. They do confirm that seeds of four invasive species are found in unburned piñon-juniper soils despite the rarity of adult plants in the undisturbed woodland. This suggests that seed banks harbor invasive species and after disturbances such as fire, these seeds are very likely immediately available for germination.

Developing a Non-native Risk Map

Soil series information was extracted using GIS for the sample points in the 1996 and 2000 fires (described above). An ANOVA was run to determine differences in the density of non-natives across soil series: significant differences were found in both data sets ($F = 3.4$, $p = .001$ for 1996 and $F = 3.9$, $p=.004$ for 2000). Invasibility of the soil series was then classified by a ordinal variable (1=low, 2=intermediate, 3=high) based upon the mean density of non-natives for that series. Unsampled soil series with similar properties were also grouped into these susceptibility classes. Vegetation communities were similarly classified (low to high susceptibility) based upon relative cover of sprouting species, characteristics of functional groups. Spatial data layers were created for susceptibility due to vegetation and susceptibility due to soils. An index of invasibility was then created using a weighted linear combination technique in the GIS in which susceptibility due to vegetation is weighted at 70 % and susceptibility due to soil fertility at 30% (Figure 5). These weightings were chosen to reflect our observations and post-fire data in which sprouting residual vegetation is the most important factor in deterring invasive species and soil characteristics, while also important, do not over-ride the vegetative effects.

DISCUSSION

Two commonly used methods were compared for their usefulness in detecting patterns of post-fire recovery in vegetation at Mesa Verde National Park. While the "Modified-Whitaker" method has worked well in many southwestern environments (Stohlgren et al. 1997), we detected the same number of species with the "survey" method in similar plot sizes. The "survey" method takes less than one-half the field time, and therefore we favor it's use for evaluating broad-scale biodiversity of native and invasive species in piñon-juniper ecosystems. The "Modified-Whitaker" method, however, allows simultaneous assessment of cover of plant species at multiple scales, which is important for other kinds of questions than the ones addressed in this study.

What patterns of biodiversity were detected in Mesa Verde? Loess habitats in the semi-arid Mesa Verde ecosystem that support a diverse native flora also tend to support a rich non-native flora, implying that high diversity habitats are also targets for non-natives. This trend was detected with the larger plot sizes only (1000 m^2 in "survey" and 100 m^2 or greater with the "Modified-Whitaker" methods). No significant trend was detected when sampling involved 1 m^2 or 10 m^2 plots. In both early post-fire (resource rich) and paired unburned habitats (possibly resource limited), a strong positive correlation was detected between the number of native species and non-native species. These data were collected primarily in deep loess soils; it should be noted that on rimrock or Mancos Shale habitats, Resource Management at Mesa Verde National Park have recorded a low incidence of non-natives (Colyer, personal communication).

In earlier studies (following the 1989 and 1996 fires) we had detected significantly greater diversity and density of invasive species in piñon-juniper habitats relative to all other vegetation communities (BAER Reports 1997, 1998, 1999). We are now able to ask questions about the mechanisms driving these patterns. Can we explain this in part by residual functional characteristics of the burned community, such as presence of sprouting species or seed banks in the pre-fire community, or by nutrient changes in burned soils?, How long do these differences persist?

To answer the first question, we surveyed points throughout the Bircher and Pony fires three years after the fire, identifying communities as woodlands or forests that lack sprouters (various piñon-juniper woodlands and Douglas-fir forests) and those with prolific sprouters (various tall shrublands and riparian areas). Despite the lack of precipitation in the years following 2000, we were able to detect a significantly greater density of invasive non-natives in those communities that lacked sprouters. Returning to the 1989 Long Mesa Fire confirmed that the invasive species persist in piñon-juniper (that lack perennial sprouter) habitats for at least 13 years. Yet there are vast differences among microhabitats in these burned habitats, witnessed by very large standard errors in density measurements. Some sites lack non-natives altogether and others virtually exclude native species by competition from very dense stands of invasives, especially muskthistle.

Next, we turned our attention to defining highly susceptible microhabitats within these habitats. We determined textural and chemical characteristics of severely burned and unburned soils. Inputs following fires significantly increased soil pH, salinity (conductivity), and the availability of nitrogen and phosphorus. As a whole these findings reflect combustion of plant materials that results in rapid mineralization and release of base-forming cations and other essential plant nutrients. Densities of non-natives within the burned areas correlated weakly with variation in soil nitrate (we believe that the correlations would strengthen if precipitation and germination were normal). Non-native densities in burned areas were also found to correlate positively with percent silt, which did not vary as a function of burning. Interestingly, in unburned sites, although non-native densities were considerably lower than in the burn, their distribution did correlate strongly with soil available phosphorus and weakly with net N mineralization and salinity. Findings from this initial analysis of the effects of fire on soil characteristics and in turn non-native densities, suggest that the availability of nitrogen and phosphorus, as well as soil textural class may be important indicators of site invasibility. (Note that these analyses excluded the recent cheatgrass invasion that began while the study was in progress).

The relative contribution of dispersal and germination may enlighten management strategies attempting to fight invasives. Therefore, we explored seed bank characteristics in burned and unburned sites. It is commonly assumed that seed banks of piñon-juniper have low diversity and density due to the exclusion of understory species over centuries of canopy development. Our data do not support this model; in fact, piñon-juniper soils contain far greater diversity of native seeds than nearby Douglas fir forests and mountain shrublands. We might expect that rapid post-fire germination of these seeds would establish native plants capable of competing with invasives. Seeds of four invasive species were found in burned and unburned piñon-juniper soils and the burned piñon-juniper soils had significantly higher density of muskthistle seeds two years after the fire. (Note, however, that this is consistent with the increases in seed-producing muskthistles in the fire the previous year; thus, we cannot determine if these seeds were present before the fire or were products of the first year post-fire muskthistle stands). Other invasive species, including *Lappula redowskii* and *Ranunculus testiculatus* were present after fires, but they were not present in the seed bank analysis. While it is possible that abundance of invasive species in piñon-juniper woodlands after fire might be influenced by a rich supply of non-native seeds in the seed bank relative to less susceptible habitats, our seed bank analysis, (carried out two years after the fire) tells us little about the relative contribution of dispersal and germination of invasives.

The final task was to represent patterns of invasibility in a spatial model that will be useful for managers to predict the most vulnerable habitats following future disturbances. The model provides a spatial representation of various risks to invasive plants based on a weighted combination of soil and vegetation. This model is preliminary and will be modified over time, but represents our best assessment of the role of functional groups, soil characteristics, and seed banks in predicting future non-native invasions with the data available at present. We will eventually expand it to the entire cuesta, however we do not have access to the nearly-completed NRSC soils map of the cuesta (Ramsey, personal communication).

In summary, we found that burned areas supported a high diversity and density of non-natives, and that piñon-juniper habitats were the most susceptible. If re-spouting species were important components of the community before fire, the likelihood of invasion was reduced. Seed banks are most diverse in piñon-juniper habitats, and they include at least four invasive species. Finally, soil type, texture and fertility, especially nitrogen, affect suceptibiltiy to invasion.

PART II: EFFECTIVENESS OF MITIGATION TREATMENTS ON INVASIVE SPECIES-- SEVEN YEARS AFTER THE CHAPIN 5 FIRE

INTRODUCTION

The Chapin 5 Fire in Mesa Verde National Park burned during August 1996, impacting seven vegetation types and hundreds of archeological sites. The Burn Area Emergency Rehabilitation (BAER) team, including the authors of this report, evaluated the damages immediately after the fire and recommended mitigation treatments to reduce erosion and prevent or lessen impacts to native vegetation recovery by noxious non-native invasion. In October 1996, Park Mesa, an old-growth piñon-juniper woodland deemed “high-risk” to non-native species was seeded with mixes of native grass seeds (Appendix A). Herbicide treatments were carried out on rhizomatous *Cirsium arvense* (Canada thistle) patches in upland areas of the Chapin 5 Fire. Mechanical removal of *Carduus nutans* (muskthistle) plants took place during the first growing season after the fire. In addition, a parallel mitigation took place, using biological control agents specific to either Canada or muskthistle. This project was carried out by Dr. Deb Kendall, Fort Lewis College, and is not part of the study reported here. BAER funding was available for two years to monitor the effects of post-fire treatments, but long-term effectiveness could not be studied under BAER funding. In this report we focus on more sustained seven and eight -year outcomes.

<u>Background Results of Earlier Monitoring with BAER Funding</u>

In 1998, there were significantly fewer muskthistle plants in seeded than in paired burned/unseeded treatments (Table 16). This trend, repeated in 1999, suggested that aerial seeding of native grass mixes was successful in reducing invasion by muskthistles. Precipitation patterns following the 1996 seeding were average for the area (Figure 3).

In 1998 we detected other noxious non-natives, including *Bromus tectorum* (cheatgrass), at densities that did not vary significantly between seeded and non-seeded areas (mean=7/75 m^2 in control and mean=4/75 m^2 in seeded, $t = 1.4$, $p > .05$). Also, *Salsola iberica* (Russian thistle) was evident in the non-seeded portions of Park Mesa (mean=16/75 m^2) but not in the seeded plots (mean=0/ m^2, $t=3.1$, $p<.05$).

Canada thistle was found in patches scattered on Park Mesa, School Section Canyon, Battleship Rock, and in Little Soda and Soda drainages. Each patch was located with GPS and sprayed with Curtail herbicide. Before and after photographs were taken of each treatment site. In 1999, points were revisited and an additional spraying took place on stands that had >20% cover.

METHODS

In 2002 and 2003, we re-sampled 80 points that had originally been established with BAER funds after the Chapin 5 Fire (Figure 2). These represented 19 burned habitats defined by vegetation/geologic substrate/slope categories. In 2002, intense drought conditions prevailed and few forbs germinated, therefore we simply recorded the presence or absence of each invasive species. In 2003 while the drought continued, adequate precipitation in May and June allowed for germination of many forbs, so we were able to record the density of each invasive species and the density of native grasses in 314 m^2 plots (10 m radius). In 2003, we also established unburned, control areas across a north to south span of the unburned portions of Chapin Mesa, simulating as much as possible the pre-fire conditions of the habitats burned during the Chapin 5 Fire.

In addition, a series of plots was established on Park Mesa in seeded and paired unseeded areas in 1997, one year after the Chapin 5 Fire. These were re-visited in 2002 and 2003; density of each native grass and invasive plant species was recorded.

2003 RESULTS

Survey of the Entire Chapin 5 Fire

Seven years after the Chapin 5 Fire, in 2003, there was a significant difference in the density of invasive species when all habitats are combined, comparing unburned and burned (untreated) points (Table 14). The most aggressive and persistent invasive species was *Carduus nutans* (muskthistle). Unburned control points had virtually no invasive species, while burned seeded and burned unseeded areas had significantly greater non-native densities. Therefore, invasive species (primarily muskthistle in this fire) persist for at least 7 years after fires.

Table 14. The density (number per 314 m2 plot, estimated for cheatgrass) of each non-native species was recorded in a series of post-fire monitoring points throughout all habitats in 2003, seven years after the Chapin 5 fire. Data are mean, standard error.

Treatment	muskthistle and herbaceous non native density	cheatgrass density	Sample size
Control, unburned	0.58 , 0.29	0.08, 0.28	48
Burned	21.9, 5.3	574.6, 178	114
Significance	F = 6.7, p =.01	F = 4.8, p= .038	

Park Mesa Seeding

How effective were aerial seeding treatments (Appendix A) after seven years? There were significantly more native grasses in the seeded plots when compared with paired unseeded plots (Table 15). This suggests that mitigation treatments provided a "jump-start" toward developing a native plant community and allowed more effective competition with invasive species. The seeded areas also had a more diverse native grass flora than unseeded. While the unseeded areas were characterized by *Poa fendleriana* (mutton grass) and *Stipa comata* (needle and thread grass), seeded areas supported *Stipa comata, Agropyron smithii* (= *Elymus smithii*, western wheat), *A. trachycaulon* (= *Elymus trachycaulus* slender wheat), *Poa fendleriana, Oryzopsis hymenoides (Indian ricegrass),* and *Sitanion hystrix* (=*Elymus elymoides*, squirrel tail grass). *Agropyron trachycaulon* is a competitive species that was selected for use on the mesa-tops to ensure grass cover although it typically occurs in canyon bottoms. We predict that the density of this species will decline over time, as it does not usually persist in an area.

Table 15. The density (number per 314 m^2 plot) of native grasses seven years after seeding treatments on Park Mesa, Mesa Verde National Park. Data are mean, standard error.

Treatment	Density	Sample size
Burned (Unseeded) control	7, 1	101
Seeded treatments	802, 49	95

F = 4.6, p <.03, significant difference among treatments

Muskthistle and other less prominent invasives occurred in higher densities in unseeded than in the seeded treatments (Table 16). This implies that, after 7 years, "natural" plant regeneration in untreated areas has been unable to out-compete invasive species. The density of invasives in the unseeded areas in 2003was nearly identical to the 1998 density, while the density of the seeded area has doubled since that year.

One of the most alarming invasive species in Mesa Verde in 2003 was cheatgrass (Figure 4, Table 17). This species has the potential, by providing continuous "flashy" ground fuels, to shorten the long fire interval native to this woodland ecosystem. While it was present in 1998 on Park Mesa, and at 5% of the Chapin 5 points 3 years after the fire, cheatgrass expanded greatly in 2003. Many researchers are attributing its prolific spread into high elevations, beyond its previous range, to climatic changes, perhaps warmer winter temperatures and effective fall precipitation in an otherwise drought cycle (Belnap, personal communication). Cheatgrass was more plentiful in burned than in unburned areas (Table 14), but in 1998 and again in 2003 we were unable to detect a difference in cheatgrass density between burned seeded and unseeded plots on Park Mesa (Table 17). Therefore, seeding mitigation has not prevented the spread of cheatgrass into burned areas.

Salsola iberica (Russian thistle) and *Lactuca* serriola (wire lettuce), both prolific invasives in 1997 and 1998, had virtually disappeared from the burned landscape by 2003. Other invasives, such as *Lappula redowski* (stick seed) and *Alyssum parviflorum* (hoary alyssum) were present in 1997 and 1998, but also were only sporatically found in 2003.

Table 16. Comparison of muskthistle density (number per hectare) in seeding treatment and control (adjacent, untreated) areas of Park Mesa, Chapin 5 Fire, Mesa Verde National Park. Data are mean, standard deviation.

Year of monitoring	Unseeded Control	Seeded Treatment	N of plots
1998	8,000 ± 728	576 ± 602 *	50
1999	13,373 ± 9301	1,779 ± 65 *	50
2001	6,125 ± 1138	3,230 ± 5933 *	26
2003	3,596 ± 7515	928 ± 1680 *	195

* ANOVA analyses for each year demonstrate significant differences (P<.05) between seeded and unseeded burned areas

Table 17. The density of *Bromus tectorum* (estimated plants per hectare) seven years after seeding treatments, Park Mesa high risk piñon-juniper habitats. Data shown are mean, standard error.

Untreated burned	Treated burned	Significance
16,913, 3962	9500, 2853	F=2.3, p=.14

Herbicide Treatments on Canada Thistle

Canada thistle patches in the upland areas of the Chapin 5 fire were treated with the herbicide Curtail in 1997 and 1998. We estimated the cover and photographed each of the upland Canada thistle stands “before” and one year “after” herbicide treatment. In 2003, six years after treatment, each area was located precisely with GPS coordinates and flagging was found in many of the stands confirming the locations. We matched pre-treatment photographs to ensure the proper site placement. The treatment was extremely effective. All of the Canada thistle was dead in five of the seven sites; in two sites, only one or two plants remained (sets of paired photos Figure 5). The treatment was highly effective in removing Canada thistle from these areas.

Mechanical Treatments on Muskthistle

Mechanical treatments took place during the first year after the Chapin 5 Fire along roadsides and in isolated early patches. Seed heads were removed and individual plants were dug out. Where native grasses such as the rhizomatous western wheatgrass were part of the post-fire successional vegetation, for example at the base of the "Tourist Trail" in Soda Canyon, this treatment was highly effective. In 2003, we established 10 plots throughout this area and no muskthistles were detected. However, where no grasses were present, this treatment was not effective and muskthistle persists today (for example on Park Mesa).

Biological control agents on both Canada and muskthistles have been successful, and their long-term effects are under investigation by other researchers and resource managers in Mesa Verde (Deb Kendall, personal communication).

DISCUSSION

Aerial seeding treatments were carried out on the MVNP Chapin 5 Fire in areas that were severely burned and were likely at risk to erosion or invasion by non-native plants. Dormant seedings of native grasses took place in October 1996, immediately following the fire. In the drought of 2002, significantly higher diversity of non-natives occurred in burned area of the Chapin 5 Fire than unburned areas. In 2003, seven years after seeding mitigation in piñon juniper sites, native grass diversity and density was significantly greater in the seeded areas and muskthistle was significantly less dense. Muskthistle remains an invasive problem in untreated areas seven years after fire. Other non-natives, including Russian thistle, alyssum, stick seed, and wire lettuce, have disappeared from the burned landscape or remain in low numbers. Recent invasions and expansions of cheatgrass into burned and unburned landscapes seem undaunted by aerial seeding treatments. We did not focus explicitly on cheatgrass because it's population increase occurred during the second and final year of this project; we strongly suggest that Resource Management and Fire personnel at Mesa Verde National Park investigate further the invasion of cheatgrass and possible effects on future fire frequency. Resource Managers have observed that cheatgrass is particularly well-established in severely burned soils directly under trees and is less dense in the inter-canopy spaces. It has been suggested that aerial seeding treatments were not as effective in the burned canopies, leaving them susceptible to cheatgrass in the subsequent years. Thus, additional treatments of the canopy spaces several years after fire might "fill-in these patches vulnerable to cheatgrass (George San Miguel, personal communication). Mechanical treatment of muskthistle stands and herbicide treatments of Canada thistle were remarkably effective; both treatments are labor intensive and must be confined to limited landscapes. While not reported here, biological control agents are effectively reducing invasive populations (D. Kendall personal communication and Mesa Verde Resource Management).

The implications of the long-term trends for management of burned piñon-juniper woodlands in Mesa Verde and other similar woodlands are profound. Recent climate changes and the increased occurrence of invasive species in the region have created a much more problematic situation than a few decades ago for managers whose goal is to promote native plant communities. Without mitigation treatments of high risk areas (those with little residual vegetation, silty soils, high nitrogen and possibly phosphorus inputs, increased conductivity, and proximity to sources of invasives), managers can expect that muskthistle will persist for at least thirteen years (as in the Long Mesa fire). Aggressive management is needed after fires, and in piñon-juniper habitats of Mesa Verde, aerial seeding of native bunch grasses, spot spraying with herbicide of early infestations, and mechanical treatments of early invasive patches reduced the invasive populations. However, invasive species cannot be eliminated, and long term solutions may include biological controls and reducing the chances of environmental disturbances in the Park. The proposed week risk model identifies particularly susceptible areas in need of mitigation following future fires and other disturbances. It is a working model that will be refined as more information, particularly about the soil texture and nutrients becomes available.

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List of Figures

Figure 1.

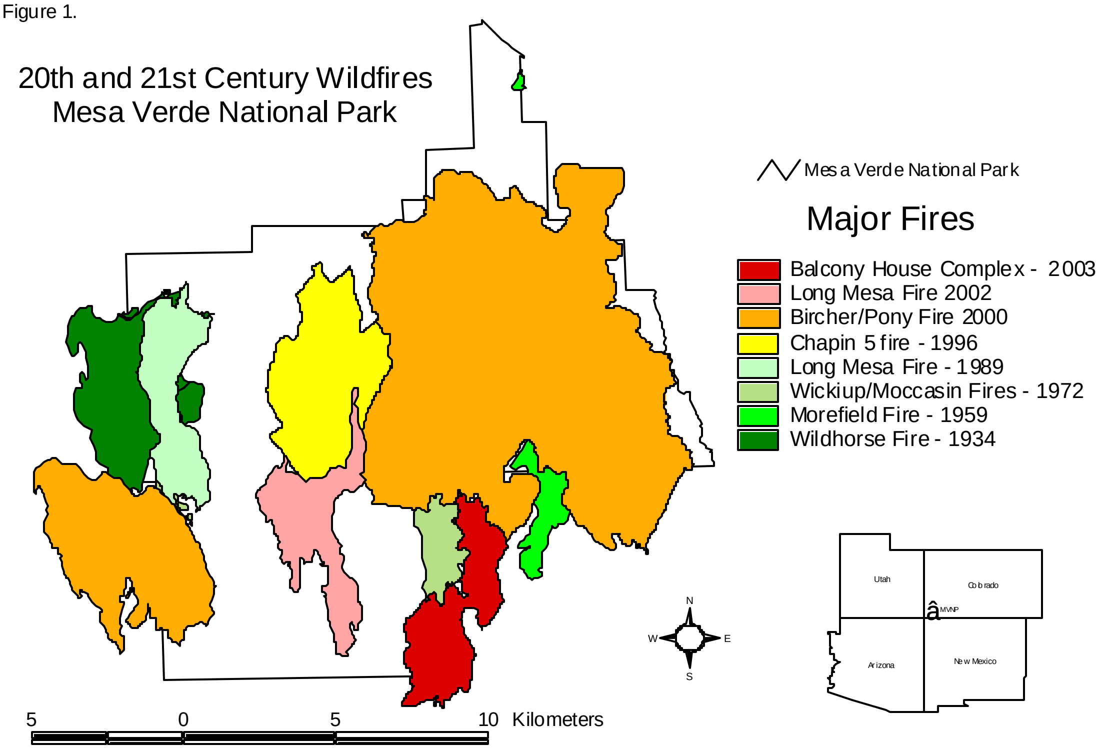

20th and 21st Century Wildfires
Mesa Verde National Park
Mesa Verde National Park
Major Fires
Balcony House Complex - 2003
Long Mesa Fire 2002
Bircher/Pony Fire 2000
Chapin 5 fire - 1996
Long Mesa Fire - 1989
Wickiup/Moccasin Fires - 1972
Morefield Fire - 1959
Wildhorse Fire - 1934
N
W
E
S
Utah
Colorado
MVNP
Arizona
New Mexico
5
0
5
10
Kilometers

Figure 2.

Sample Locations

1989 Long Mesa, 1996 Chapin 5 and the 2002 Bircher and Pony Fires

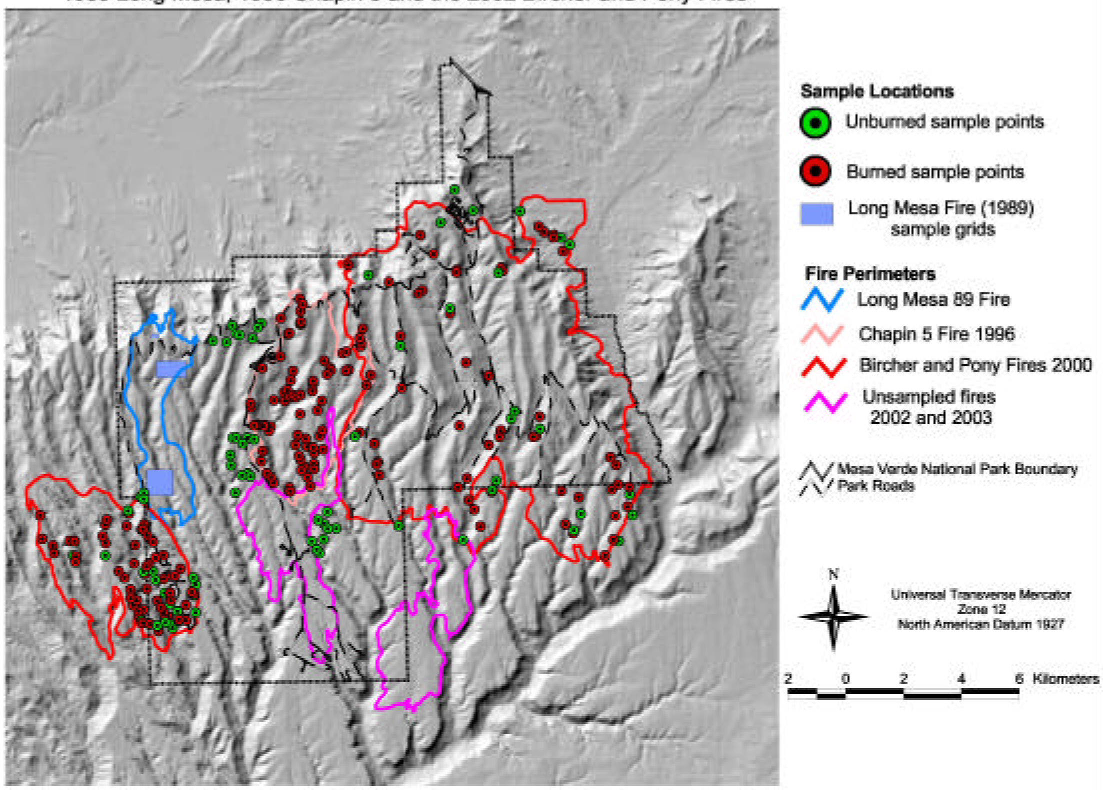


Figure 3.

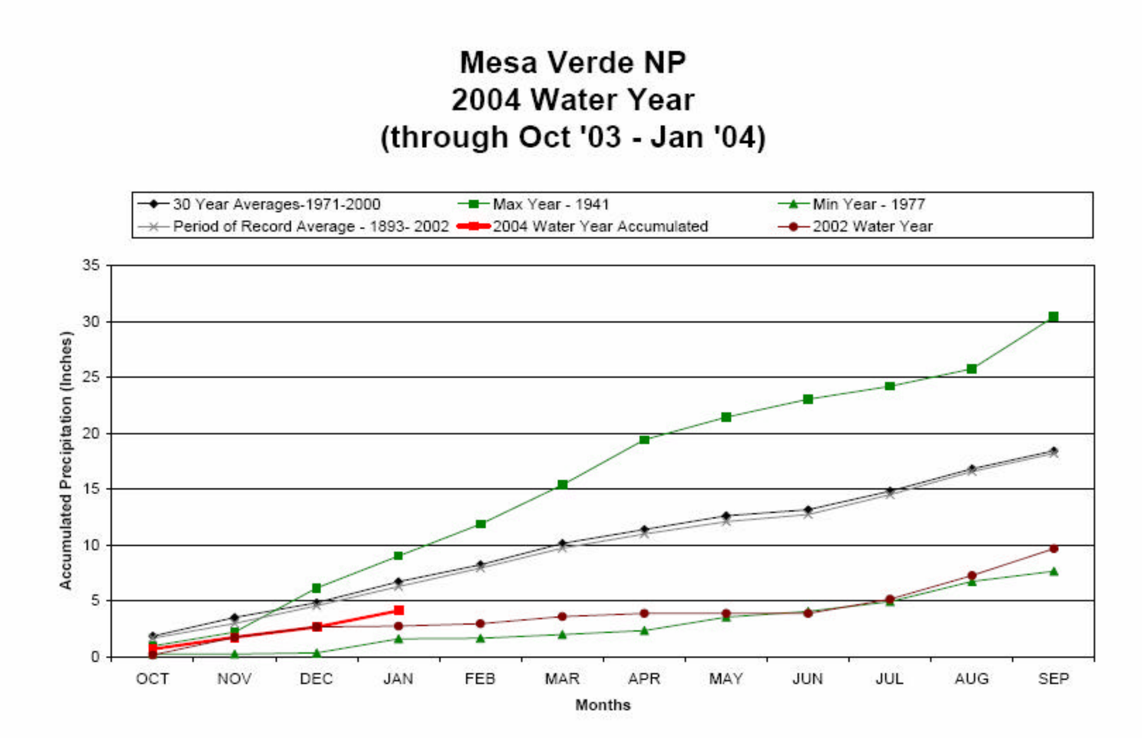
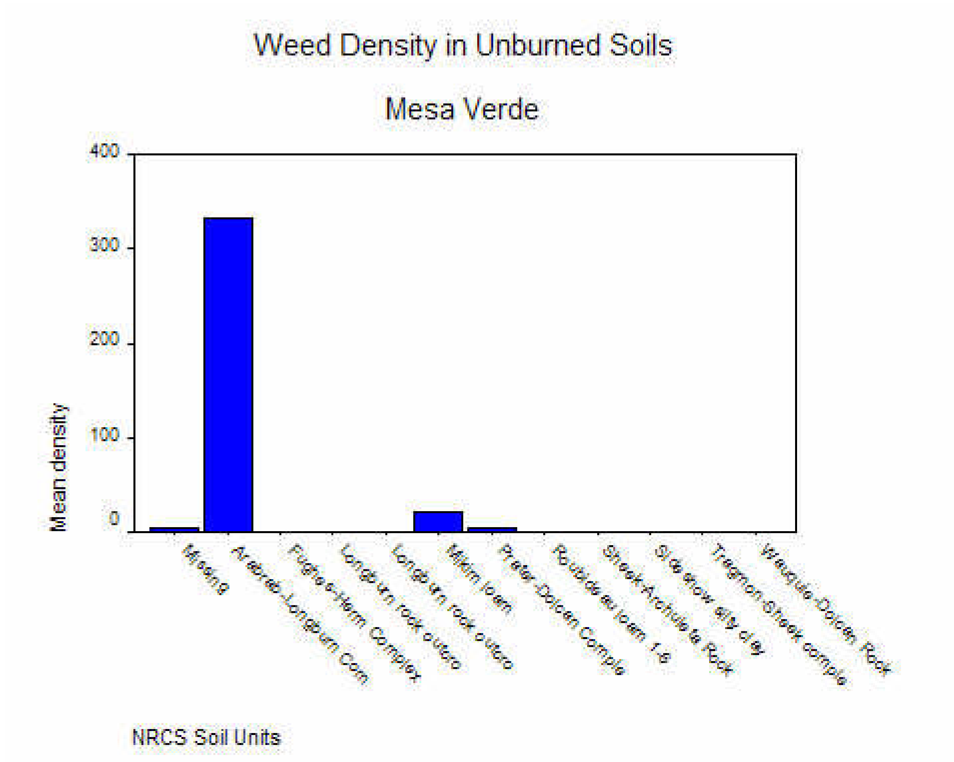
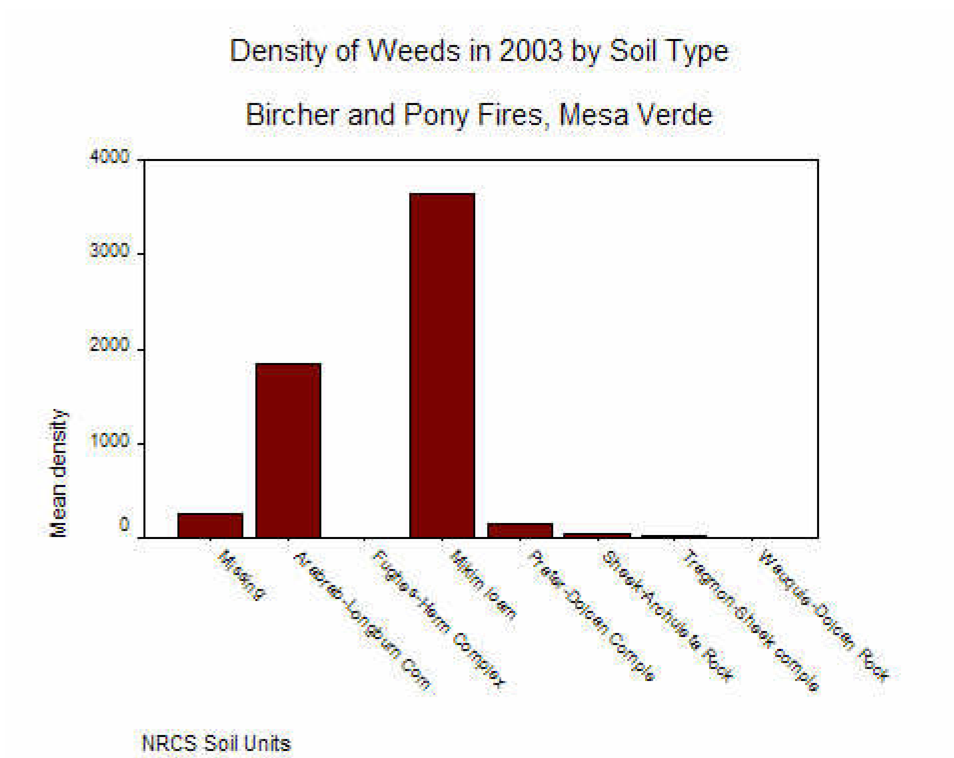


Figure 3. Precipitation trends in 1941 and 1977 (wet years) are contrasted with recent trends in precipitation on Mesa Verde. These data document drought conditions since 1997. (Colorado Climate Center, http://ccc.atmos.colostate.edu/pdfs/mesaverdeWY2004.pdf)

Figure 4


Weed Density in Unburned Soils
Mesa Verde
400
300
200
100
0
Mean density
Missing
Mikim loam
NRCS Soil Units


Density of Weeds in 2003 by Soil Type
Bircher and Pony Fires, Mesa Verde
4000
3000
2000
1000
0
Mean density
Missing
Mikim loam
NRCS Soil Units

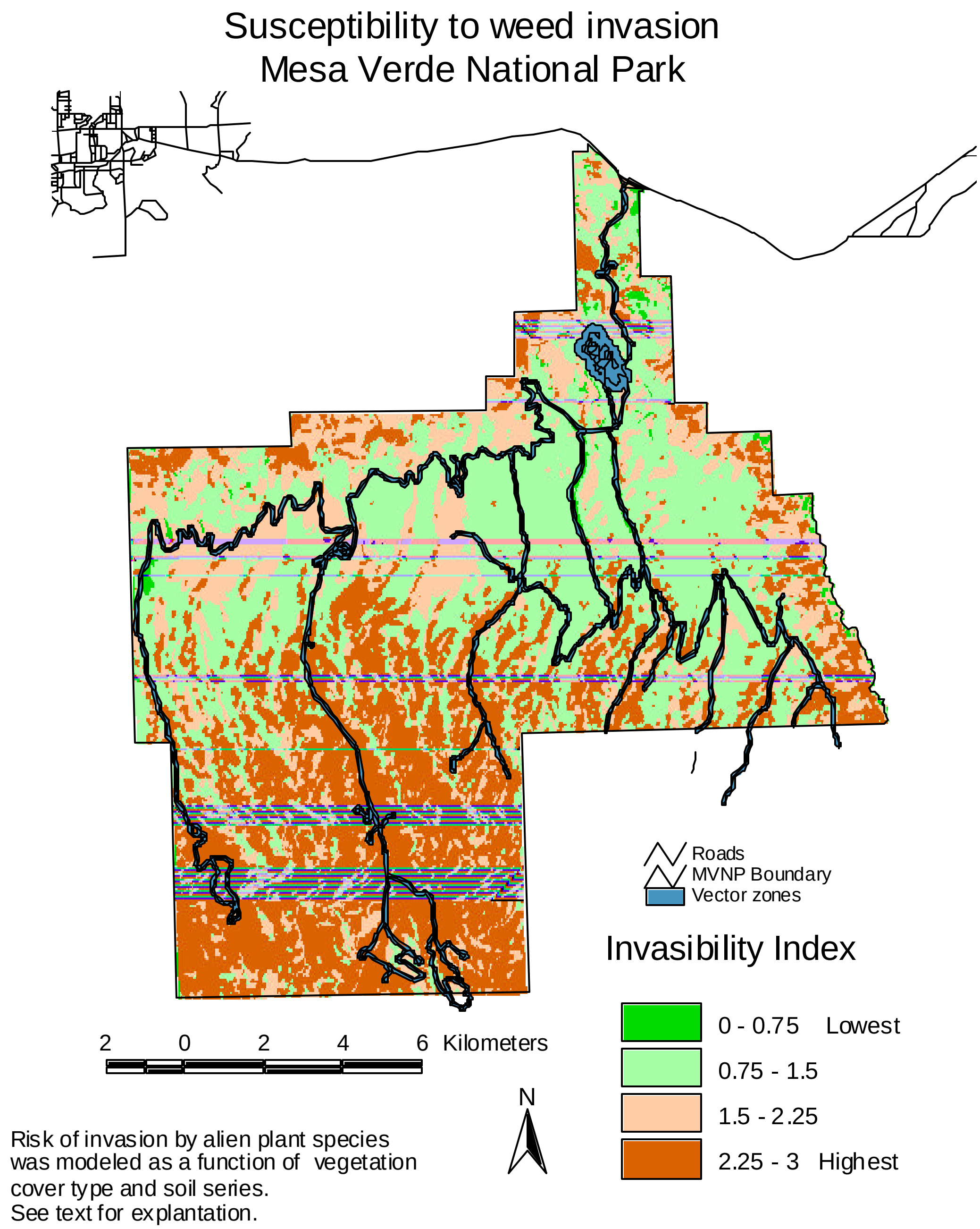

Susceptibility to weed invasion
Mesa Verde National Park
Roads
MVNP Boundary
Vector zones
Invasibility Index
0 - 0.75 Lowest
0.75 - 1.5
1.5 - 2.25
2.25 - 3 Highest
2 0 2 4 6 Kilometers
N
Risk of invasion by alien plant species
was modeled as a function of vegetation
cover type and soil series.
See text for explantation.

Figure 6. Cheatgrass (*Bromus tectorum*), Park Mesa, Mesa Verde National Park. June 2003

Results of the chemical Treatments on Canada Thistle (*Cirsium arvense*) after the Chapin 5 fire.

31 August, 1998

17 June 2003

All photos are at site #10.

Results of the chemical Treatments on Canada Thistle (*Cirsium arvense*) after the Chapin 5 fire.

15 August, 1998

31 August, 1998

17 June 2003

All photos are at site #11

Appendix A:

1996 Chapin 5 Fire

Fire	Acres seeded	Species	Proportion of mix
Chapin 5 Fire- north Park Mesa	77	*Kohleria cristata* *Oryzopsis hymenoides* *Agropyron trachycaulum* *Sitanion hystrix*	.10 .20 .30 .30
Chapin 5 Fire- south Park Mesa	201	*Poa fendleriana* *Oryzopsis hymenoides,* *Agropyron trachycaulum* *Sitanion hystrix*	.10 .20 .30 .30

2000 Bircher and Pony Fires

Mix Type	Acres seeded	Species	Proportion of Mix
Mix 1: shale slopes	109	*Bouteloua gracilis* *Agropyron smithii* *Hilaria jamesii* *Sitanion hystrix*	.25 .25 .25 .25
Mix 2: NPS, Northern mesas	724	Koehleria cristata Oryzopsis hymenoides *Agropyron trachycaulum* *Agropyron smithii* *Sitanion hystrix*	.20 .20 .30 .20 .10
Mix 3: southern mesas eastern (Bircher), Wetherill Mesa southern (Pony)	2745	*Poa fendleriana* *Oryzopsis hymenoides* *Agropyron trachycaulum* *Agropyron smithii* *Sitanion hytrix* *Stipa comata* *Stipa viridis* *Bromus anomola*	.16 .20 .20 .10 .15 .20 .05 .04
Mix 4: UMU mesa tops	1574	*Agropyron trachycaulum* *Festuca arizonica* *Oryzopsis hymenoides* *Poa sandbergii*	.40 .40 .16 .04
Mix 5: BLM mesa tops	241	*Agropyron trachycaulum* *Bouteloua gracilis* *Oryzopsis hymenoides*	.30 .30 .40
Mix 6: S. canyons NPS, and Rock Canyon (Pony)	2885	*Oryzopsis hymenoides* *Agropyron trachycaulum* *Agropyron smithii* *Sitanion hystrix*	.20 .30 .20 .10
Mix 7: Pony mesa tops UMU	856	*Poa sandbergii* *Oryzopsis hymenoides* *Bouteloua gracilis/Sporobolus*	.10 .30 .60

		cryptandrus	